Density Stratification in Onondaga Lake: 1968-1994[1]

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ABSTRACT

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The profound impact of ionic (saline) waste from soda ash production on the density of inflows to, and the stratification regime of, Onondaga Lake, NY is documented. The analysis is based on inflow data reported by Effler et al. 1996 in this issue and on lake monitoring data collected over the 1968-1994 interval. Particular emphasis is placed on characterization of changes that followed closure of the soda ash manufacturing facility. Inflows enriched with the ionic waste have been more dense than the lake surface and entered as underflows, sometimes plunging to the lake bottom. Density differences have been reduced, but not eliminated, following closure of the facility. During the operation of the facility, the lake annually had a significant salinity component of density stratification, spring turnover failed to occur in a number of years, the duration of summer stratification was extended, and reformation of salinity-based stratification in fall was common. A nearly 90% reduction in ionic waste loading from soda ash production has substantially ameliorated these impacts, though some impact continues to be observed from the residual loading. The impacts of ionic waste on the lake's stratification regime have exacerbated the lake's problem of limited hypolimnetic oxygen resources.

Key Words: density stratification, salinity, underflow, turnover, dissolved oxygen, ionic waste.

Thermal stratification is a ubiquitous phenomenon in deep lakes in temperate climates (e.g., Hutchinson 1957, Wetzel 1983), and is an important regulator of the overall metabolism of these systems. Features of the stratification regime, such as its interplay with mixing, the vertical dimensions of the layers, the temperatures of the layers, and the duration of stratification, mediate the cycling of key materials (e.g., nutrients), primary production, rates of biochemical reactions, and the oxygen resources of hypolimnia (e.g., Bowie et al. 1985, DiToro and Connolly 1980, Lam and Schertzer 1987, Martin et al. 1985, Owens and Effler 1989, Powell and Jassby 1974, Wodka et al. 1983). The important dependence of various measures of lake water quality on stratification has been noted by a number of investigators (e.g., Effler 1987, Orlob 1983, Owens and Effler 1989, Stefan et al. 1976, Stauffer and Lee 1973). Thus, modification of the natural stratification/mixing regime of a lake by anthropogenic inputs has ecological and water quality implications. Inflows made dense from anthropogenic ionic enrichment have been observed to enter receiving lakes as plunging underflows (Effler and Owens 1986, Fischer and Smith 1983, Wetzel 1983), and thereby modify the stratification regime (e.g., Effler et al. 1986a).

Here we describe density differences that have prevailed between inflows to Onondaga Lake, NY, and the lake, and related impacts to the stratification regime of the lake, that are largely attributable to the discharge of saline (i.e., dense) waste from soda ash production. Salient features of the lake, including its inflows and degraded state, and the history of operation and ionic waste discharges of a soda ash/chlor-alkali facility, have been described in other manuscripts in this issue (Effler et al. 1996, Effler and Hennigan 1996). This analysis is based on characterizations of the ionic content of the inflows presented in an accompanying manuscript (Effler et al. 1996), and a long-term data base of temperature and Cl^- concentration profiles for the lake. We describe the occurrence and seasonality of

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salinity stratification in the lake and the importance of meteorological variability in mediating year-to-year differences in the phenomenon. Emphasis is given to identifying changes that have occurred as a result of a major decrease in ionic waste loading to the lake associated with the closure (1986) of the soda ash/ chlor-alkali facility (Effler et al. 1996). Additionally, we describe the exacerbating effect that alterations in the stratification regime of the lake, caused by the ionic waste inputs, have had on the lake's problem of degraded oxygen resources.

Methods

Estimates of density were based on paired measurements of temperature and Cl^- concentration. The concentrations of Cl^- has been demonstrated to be a reliable estimator of salinity for both the period before closure of the facility (Effler et al. 1986c) and after closure (Effler 1996). The supporting tributary monitoring program was described by Effler et al. (1996). The characterization of the stratification regime of Onondaga Lake presented here is based on profiles of temperature and Cl^- concentration collected at a deep (~19 m) location in the lake's southern basin. Owens and Effler (1996b) concluded, based on analysis of more spatially complete monitoring data (Field 1980, Stewart 1971), that stratification conditions at this site were generally representative of lake-wide conditions. Data for the pre-1980 period were collected by Onondaga County (Onondaga County 1971-1980). Profiles from that program usually were obtained at 3 m intervals, bi-weekly, from the spring through fall period. Data after 1980 are from a more intensive monitoring program in which profiles of T and Cl^- were collected at 1 m depth intervals, at a frequency of 2-3 times per week in 1980 and 1981, and weekly thereafter until 1988. Since 1989, the vertical resolution of Cl^- monitoring has been reduced (e.g., 16 out of 20 depths measured).

Chloride was measured by the $Hg(NO_3)_2$ titration method (American Public Health Associated 1980). Temperature was measured with a Montedoro-Whitney thermistor (Model TC-5C, sensitivity of ± 0.05°C) through 1990, and with a HYDROLAB (Surveyor 3) thereafter. Dissolved oxygen (DO) measurements were made with calibrated instrumentation at 1 m intervals; with a YSI (Model 54) DO meter from 1978 through 1990, and with the HYDROLAB (Surveyor 3) thereafter.

Estimates of density were made with an equation of state (Effler 1996) that incorporates the density - temperature relationship for pure water presented by Millero et al. (1976) and the salinity dependence developed by Chen and Millero (1978). This relationship performs equally as well as the system specific expression (Effler et al. 1986c), for the density difference (e.g., stratification) issues addressed here. The densities of selected inflows relative to the lake were represented as differences between the density of the inflow and the density of the lake surface; positive values indicate a tendency for the inflow to plunge in the lake (enter as an underflow; Effler and Owens 1986). Lake surface densities were temporally interpolated, for cases in which the days of tributary and lake monitoring differed. Vertical density differences at 1 m intervals ($g/cm^3/m$) and for the entire water column were calculated for the lake, and the contributions of salinity and temperature to density were evaluated, according to the procedures set forth by Effler et al. (1986c). These procedures have previously been applied to the data sets for 1980 (Effler et al. 1986a) and 1981 (Owens and Effler 1989).

Density Differences Between Inflows and the Lake

Seasonal density differences between inflows and the lake are only considered for the three largest inputs, Ninemile Creek, Onondaga Creek, and the Metropolitan Sewage Treatment Plant (METRO) (Fig. 1). Systematic changes in density difference occurred as a result of shifting the soda ash/chlor-alkali facility discharge from Ninemile Creek to METRO (starting May of 1981 and extending to early 1986; described by Effler et al. 1996), and by the closure of the facility in 1986 (Fig. 1). The temporal details for the tributaries are strongly influenced by the time structure of runoff (e.g., snowmelt and rainfall events), which vary greatly year-to-year in this region. However, selected periods serve to illustrate important inflow characteristics and conspicuous temporal changes. The ionically enriched tributaries (Effler et al. 1996) tend to be relatively more dense in late summer (Fig. 1a and c) when natural inflow (i.e., dilution) is low. Diversion of the ionic waste to METRO reduced the density (and thus negative buoyancy) of the Ninemile Creek inflow (Fig. 1; i.e., tendency to plunge reduced), but made the METRO effluent distinctly more dense (and negatively buoyant; Fig. 1b). Thus both of these inflows tended to plunge in the lake during the 1981-1986 period when ionic waste was diverted to METRO.

The density differences between Ninemile Creek and the lake did not change greatly from the period of active (e.g., 1981-1986) diversion to post-closure (Fig. 1a), despite further reductions in the ionic content of

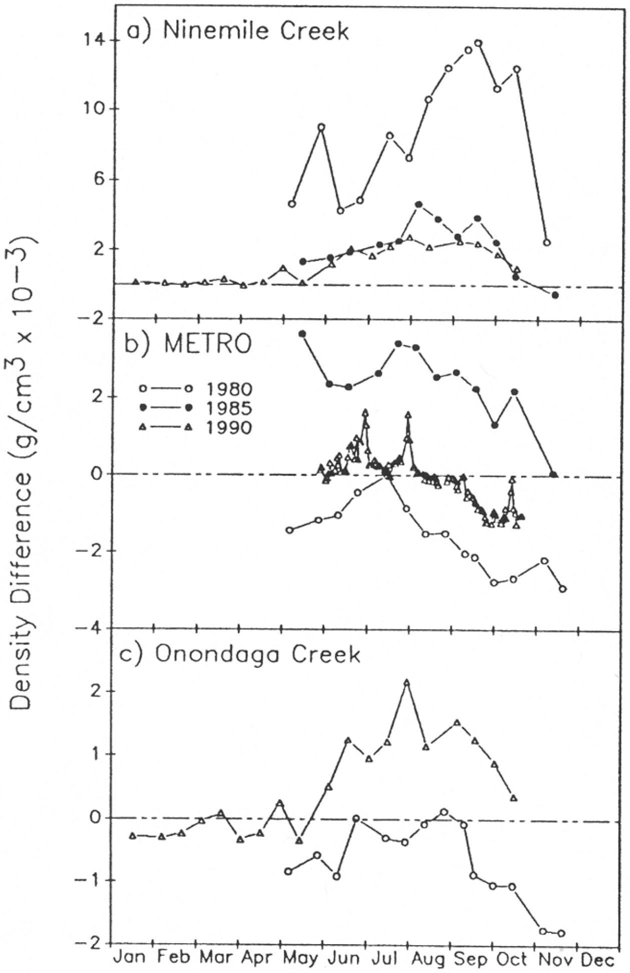

Figure 1.–Density differences between inflows and the surface waters of Onondaga Lake, for selected years: a) Ninemile Creek, b) METRO, and c) Onondaga Creek. Positive density difference indicates the inflow is more dense than the lake surface (i.e., negatively buoyant) and tends to plunge.

the stream over that interval (Effler et al. 1996). This reflects the compensating effects of a reduction in salinity of the lake over the same period (e.g., Doerr et al. 1994, Effler et al. 1996), and the continuing input, albiet reduced, of ionic waste into Ninemile Creek (Effler et al. 1991, 1996). The METRO effluent tended to enter as a buoyant (i.e., less dense) overflow before reception of the ionic waste (Fig. 1b). This discharge has usually been approximately neutrally buoyant since the closure of the soda ash/chlor-alkali facility, though the irregular reception of ionic waste (Effler et al. 1996) at times makes it distinctly more dense (i.e., negatively buoyant; e.g., see Fig. 1b for 1990). Onondaga Creek was less dense than the lake for most of the year before closure (Fig. 1c). However, it is often more dense than the lake since closure (Fig. 1c) because of the reduction in the salinity of the lake (Doerr et al. 1994, Effler et al. 1996).

Salinity Stratification

Monthly Cl^- concentration profiles over the April-October interval of four years (Fig. 2) illustrate important cases of the occurrence and seasonality of salinity stratification in Onondaga Lake. Two years from before (Fig. 2a-g) and after (Fig. 2h-n) closure of the soda ash/chlor-alkali facility were selected to illustrate not only the strong differences between the pre-closure and post-closure conditions, but also the significant year-to-year variations associated with meteorological variability. Chloride concentrations were much higher and salinity stratification was much greater before closure, but the underflow phenomenon (Effler and Owens 1986) occurred in all the years (Fig. 2). It is important to note that the scaling for Cl^- concentration for the post-closure years has been expanded (Fig. 2h-n) to better show the continuing manifestations of the underflow phenomenon for the enriched inflow(s). Data from one year before (1980) and after closure (1993) illustrate the failure of spring turnover that resulted from plunging underflows (Effler and Perkins 1987). The other two years illustrate the case of the occurrence of spring turnover.

The lake can be described as having a monomictic stratification regime in years in which spring turnover failed to occur, in contrast to other lakes of similar size in the region that have a dimictic regime. A monomictic stratification regime prevailed in 1980, and was manifested in many other years before closure (Effler and Perkins 1987). Salinity stratification was well established by mid-April in 1980 (Fig. 2a). Wind-induced mixing was inadequate in that year to break up the salinity-based density stratification that was established by the plunging underflow of ionically-enriched Ninemile Creek during the antecedent ice-cover period (Owens and Effler 1989). Model simulations (Owens and Effler 1989) and available monitoring data indicate fall turnover probably occurred in all years before closure of the soda ash/chlor-alkali facility, though it was generally delayed (e.g., Fig. 2g) compared to other local lakes, and interrupted by temporary reformation of stratification.

In contrast to 1980, Onondaga Lake exhibited a dimictic stratification regime in 1981 (Fig. 2a-g); spring turnover was evident in mid-April as a vertically uniform Cl^- profile. The strong difference in salinity stratification in spring of the two years (Fig. 2a) is consistent with the much greater wind energy input to the lake in spring of 1981 compared to 1980 (Effler et al. 1986b). Salinity stratification was established by mid-May of 1981 (Fig. 2b), indicating the subsequent entry of the plunging saline underflow into lower layers. This is consistent with the greater propensity of a dense inflow to plunge

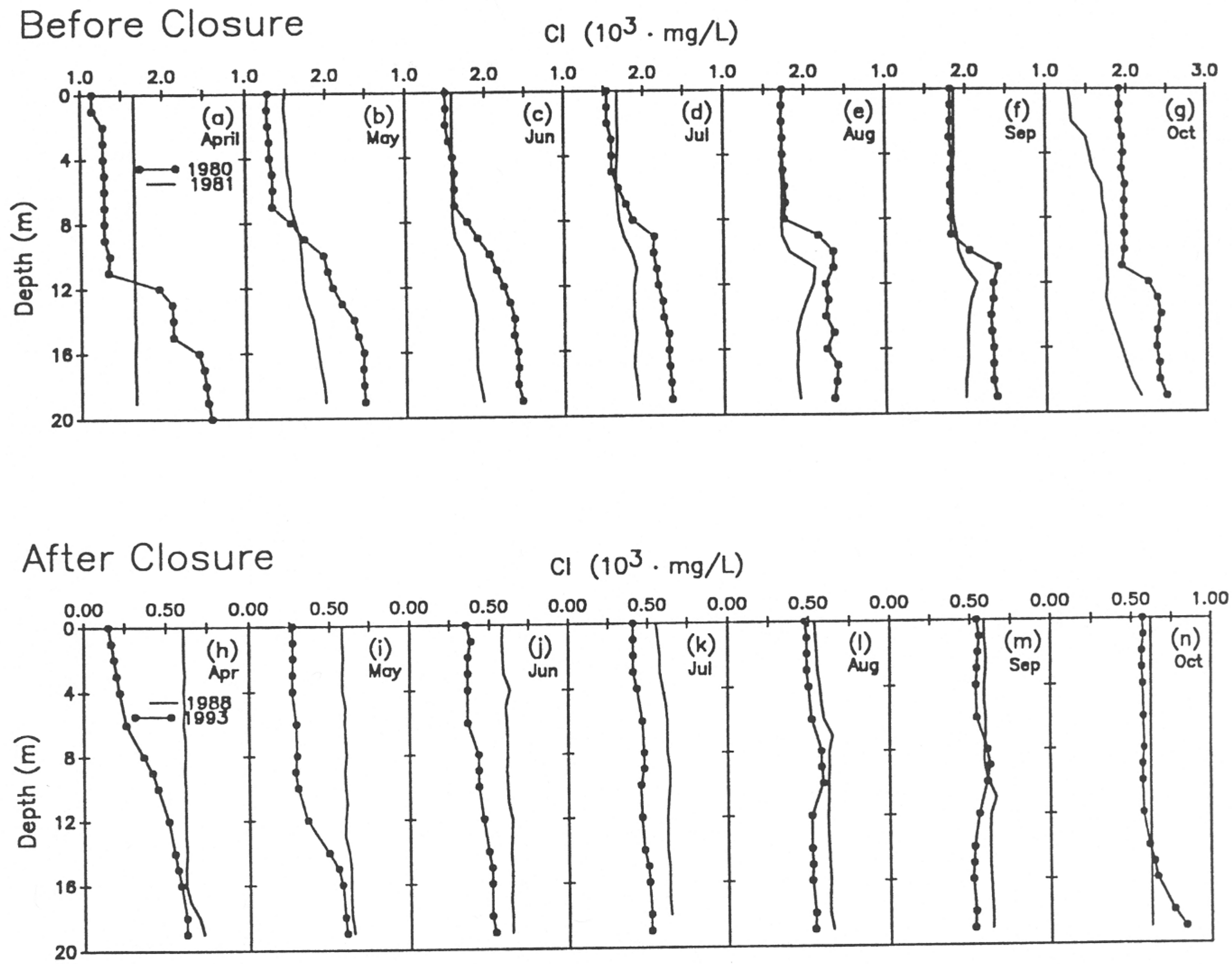

Figure 2.–Mid-month Cl⁻ profiles for Onondaga Lake for selected years: a) April, 1980 and 1981, b) May 1980 and 1981, c) June, 1980 and 1981, d) July, 1980 and 1981, e) August, 1980 and 1981, f) September, 1980 and 1981, g) October, 1980 and 1981, h) April, 1988 and 1993, i) May, 1988 and 1993, j) June 1988 and 1993, k) July, 1988 and 1993, l) August, 1988 and 1993, m) September, 1988 and 1993, and n) October, 1988 and 1993.

when the stability (density stratification) of the lake's water column is minimal (Owens and Effler 1989, 1996a). The extent of salinity stratification remained lower through the summer of 1981 compared to 1980 (Fig. 2a-g) as a result of the occurrence of spring turnover in 1981.

In the years before closure, the dense, ionically-enriched inflow(s) tended to enter the water column above the thermocline (i.e., after thermally-based stratification was established) because it could not penetrate the density gradient (Effler et al. 1986a, Owens and Effler 1989). At times this produced modest mid-depth maxima (interflows) in Cl^- concentration (e.g., Fig. 2d, e, and f), as the chemocline was positioned slightly above the thermocline (e.g., Effler et al. 1986a). The routing of ionically enriched inflow(s) to the epilimnion after establishment of substantial density stratification resulted in progressive increases in the Cl^- concentration of the epilimnion, and thereby progressively decreased the extent of salinity stratification (Fig. 2).

The 1988 data (Fig. 2h-n), and monitoring results from the 1987-1992 interval, indicate that closure of the soda ash/chlor-alkali facility has greatly ameliorated the impact of ionic waste discharge on the lake's stratification regime, as a result of the attendant reductions in the density differences between the lake and its inflows (Fig. 1). For example, the extent of salinity stratification in 1988 was greatly reduced (Fig. 2h-n) compared to conditions before closure. However, subtle indications of plunging underflows persisted, as indicated by increases in Cl^- concentration in the lowermost layers in mid-April (Fig. 2h) and minor mid-depth increases in Cl^- observed in mid-August (Fig. 2l), and mid-September (Fig. 2m).

The 1993 observations (Fig. 2h and i) indicate that continued negative buoyancy of saline inflows (particularly Ninemile Creek, Fig. 1a) can result in a degree of salinity stratification in spring that causes the failure of spring turnover (i.e., monomixis). In 1994, salinity stratification that developed under the ice again prevented spring turnover. These most recent failures of spring turnover are not attributable to increases in ionic pollutant loading. Rather they reflect the lingering effects of the continuing saline inputs (Fig. 1a, Effler et al. 1996) that are manifested only under certain meteorological conditions. For example, runoff received in April 1993 was the highest recorded in the last 40 y. This further enhanced the antecedent salinity stratification by reducing the Cl^- concentration of the upper waters (dilution). Additionally, ice cover persisted later than usual (e.g., early April instead of early March; personal communication, K. Stewart) into the springs of 1993 and 1994, thereby reducing the period of exposure to wind mixing in spring. Plunging to mid-depths is indicated during the summer, and to the bottom (Fig. 2n) in October, of 1993.

Density Stratification, Salinity and Thermal Components

To resolve the contributions of salinity and temperature to overall density stratification we performed detailed analysis of paired vertical profiles of Cl^- (i.e., salinity) and temperature (Fig. 3). The analysis illustrated for July 30, 1980 (Fig. 3) was conducted for each set (2) of profiles. For example, Onondaga Lake

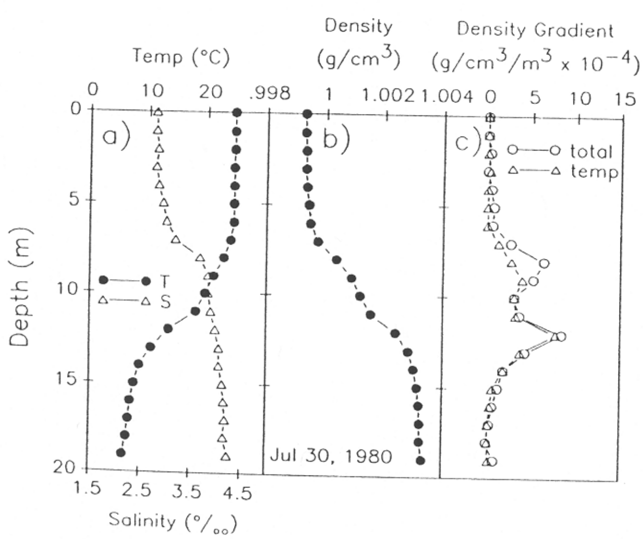

Figure 3.–Density stratification in Onondaga Lake; vertical resolution of salinity and thermal components for July 30, 1980.

had well developed salinity and thermal stratification on July 20, 1980 (Fig. 3a). The chemocline was displaced above the thermocline in the water column, consistent with the behavior of a dense saline inflow for strongly stratified periods described earlier. Density differences were largely located between 6 and 14 m (Fig. 3b) on this day. The salinity component dominated the overall density gradient between 7 and 8 m, and represented about 27% of the overall (i.e., top to bottom) vertical density difference at the time of measurements (Fig. 3c). Subsequent presentations of seasonality are consolidated forms of these analyses conducted for many individual days.

The seasonal partitioning of density stratification into thermal and salinity components is presented here for 15 years over the 1968-1994 interval (Fig. 4). The analysis depicts the relative role of salinity stratification, changes since closure of the soda ash/ chlor-alkali facility, and the extent of interannual variability. There was a significant salinity component of overall density stratification annually in Onondaga Lake before closure (Fig. 4a-g). Further, the salinity component of stratification decreased in a nearly linear manner during summer before closure. Progressive decreases in the salinity component of density stratification were also observed in 1993 (Fig. 4n) and 1994 (Fig. 4o). These are the only two years since closure that this component of stratification has been observed in the spring.

The salinity component of stratification represented from 18% (1985, Fig. 4f) to 50% (1972, Fig. 4b) of overall density stratification over the May-September interval for the pre-closure years considered. Interannual variations in the contribution of the salinity component are attributed largely to the influence of meteorological variability. In 1993, the year of maximum salinity stratification since closure (Fig. 4n), the salinity component represented only 14% of the total over the same months. There was no significant salinity component during these months over the 1987-1992 period. The year of closure of the soda ash facility (1986) represents a special case, due to the abrupt reduction in ionic waste loading and the rather rapid flushing of the upper lake waters with more dilute inflow (Doerr et al. 1994). This tended to maintain the salinity component throughout the summer period (Fig. 4g) because reductions in salinity stratification due to vertical mixing were compensated by continuing reductions in epilimnetic salinity (associated with the reduced salinity of inflows).

The effect of plunging saline inflow(s) on the establishment or reestablishment of salinity stratification in the lake is particularly well depicted in certain of the years included in the analysis (Fig. 4). The development of salinity-based density stratification

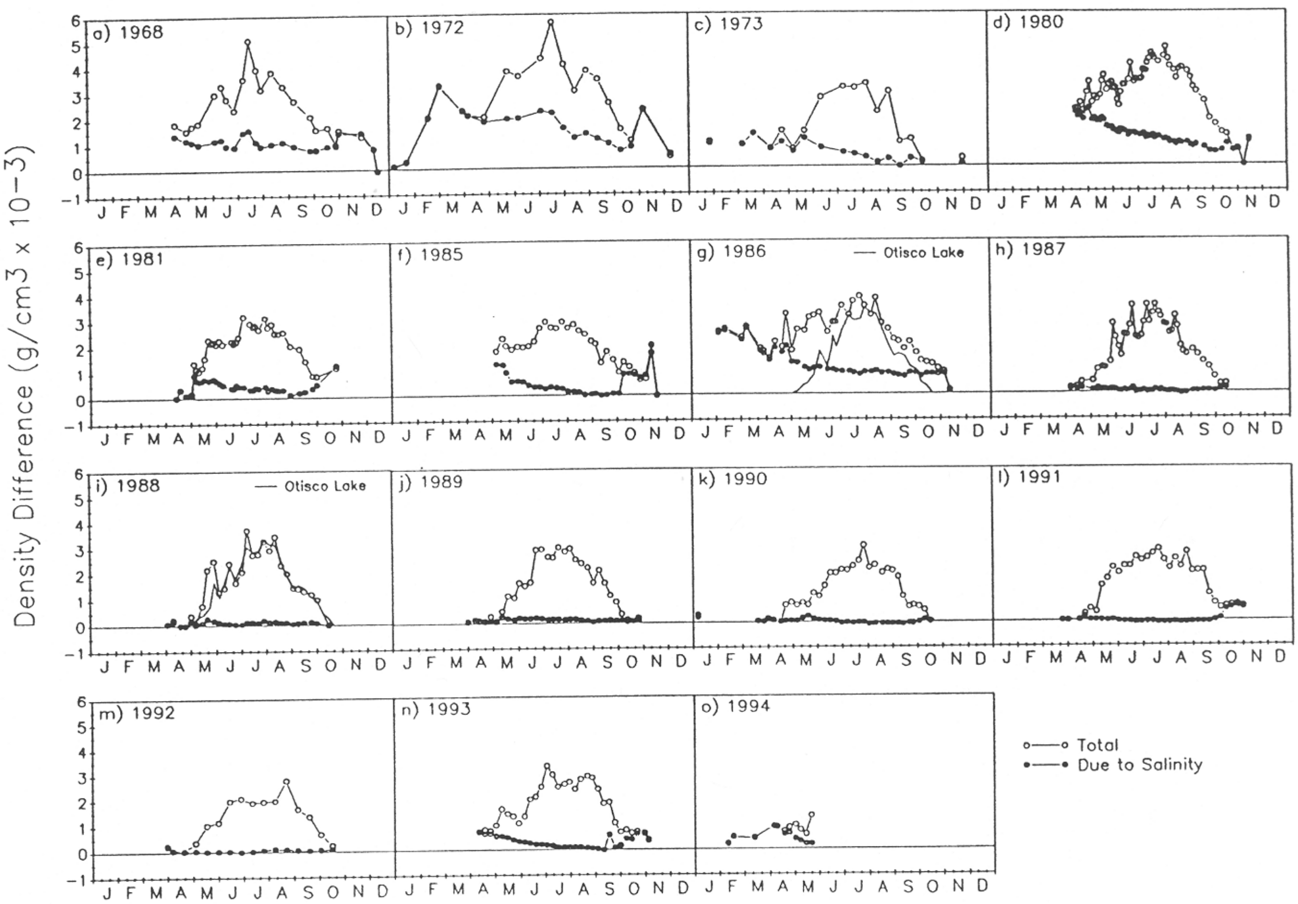

Figure 4.–Temporal distributions of total (i.e., bottom minus top) watercolumn (for maximum depth site, 19 m) differences in density, and the S component of density stratification in Onondaga Lake: a) 1968, b) 1972, c) 1973, d) 1980, e) 1981, f) 1985, g) 1986, compared to Otisco Lake observations, h) 1987, i) 1988, compared to Otisco Lake observations, j) 1989, k) 1990, l) 1991, m) 1992, n) 1993, and o) 1994.

under the ice is illustrated by wintertime measurements in 1972 (Fig. 4b) and 1994 (Fig. 4o). The unusually long (e.g., compared to other local lakes) persistence of density stratification in fall to early winter, associated with the salinity component, was manifested in most years before, and the year of, closure (Fig. 4). The reformation of salinity-based density stratification caused by the plunging inflow(s) was observed in most years before closure (Fig. 4). This phenomenon has continued since closure (e.g., Fig. 4l and n), albeit less intensely and perhaps less frequently because of the reduction in density differences between the inflows and the lake (Fig. 1). Extension of monitoring later into fall and early winter probably would have depicted the interruption of turnover in all the monitored years before and after closure.

The occurrence of salinity stratification in Onondaga Lake has impacted not only overall density stratification in the lake but also the thermal component. For example, note the disparity in the thermal component in Onondaga Lake and thermal (total) stratification in nearby (22 km) Otisco Lake in 1986 (Fig. 4g), a year in which the salinity component was important in Onondaga Lake. Yet the stratification regimes of these two lakes were quite similar in 1988 (Fig. 4i), when the salinity component (underflow phenomenon) in Onondaga Lake was insignificant. The failure of spring turnover, induced from saline underflow(s), resulted in unusually low bottom water temperature (Fig. 5). Examples include 1980 (before closure, Fig. 5a), 1986 (year of closure, Fig. 5a), 1993, and 1994 (after closure, Fig. 5b). In all these cases bottom temperatures were distinctly lower than observed for years in which the lake experienced the rapid heating that attends spring turnover. Subsequent periods (e.g., May) of rapid heating (i.e., mixing) in 1986 (Fig. 5a) and 1994 (Fig. 5b) were inadequate to produce the temperatures observed in years in which spring turnover occurred. The progressive warming of the bottom waters of Onondaga Lake during the period of summer stratification in all years (Fig. 5), also observed in other lakes (e.g., Jassby and Powell 1975), is a result of limited mixing with the overlying warmer lake layers.

Impact on Oxygen Resources

Dissolved oxygen (DO) is lost rapidly from the lake's hypolimnion because of its highly eutrophic state (Effler et al. 1986b). There is compelling evidence that the ionic waste from soda ash production has exacerbated the lake's problem of limited oxygen resources, and that lingering impacts continue because of the continued (albeit reduced; Effler et al. 1996) loading of this waste. When spring turnover fails to occur, the lower layers of the lake are not replenished with DO. As a result, anoxia has been observed in the lowermost waters in the early spring of these years. This anoxia has persisted and expanded (vertically) through summer (e.g., Effler and Perkins 1987). The reformation of salinity-based density stratification during the fall mixing period further impacts the oxygen resources of the lake by temporarily isolating the bottom layers from the overlying oxygenated water. This has been, and continues to be, manifested as abrupt depletions of DO in the lowermost layers (Fig. 6a-c). Each of the examples of this phenomenon presented (Fig. 6a-c) correspond to the transition from turnover conditions to the reformation of stratification

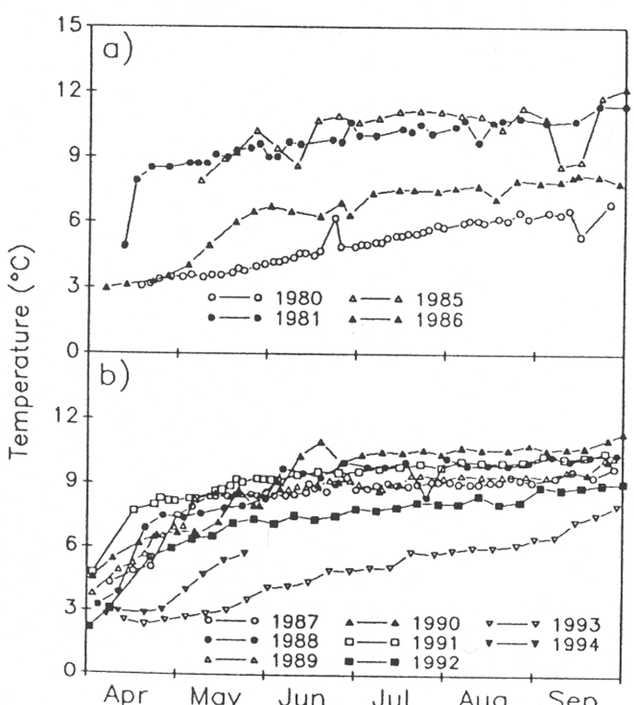

Figure 5.–Time series of bottom water (19 m) temperatures in Onondaga Lake, for selected years: a) before, and the year of, closure of the soda ash facility, and b) after closure of the soda ash facility.

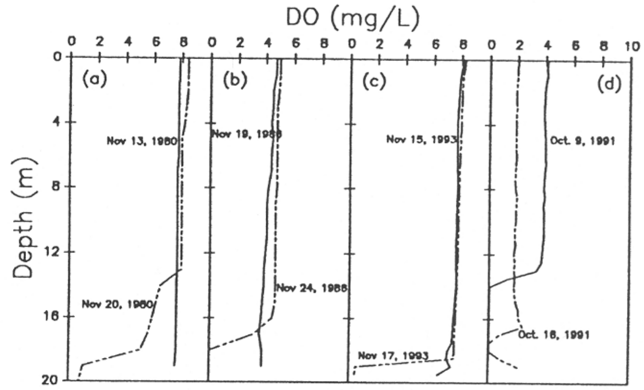

Figure 6.–Examples of rapid DO transformations in the bottom waters of Onondaga Lake in the fall, coupled to reformation of density stratification by saline underflow: a) 1980, b) 1986, c) 1993, and d) 1991.

over a short (≤ 7d) interval. Irregularly, with the approach to fall turnover, conspicuously high DO concentrations have been observed in the lowermost layer, as a result of plunging of well oxygenated saline inflows (e.g., 6d).

Effler (1987) speculated that, before closure, prolonged stratification associated with the saline discharge from soda ash production extended the period of anoxia, and thereby exacerbated the lake's problem of limited oxygen resources. The temporal distributions of the upper bound of anoxia in the lake before (and the year of) and after closure (Fig. 7) supports this position. The pre-closure data are partitioned into the "failure of spring turnover" case (Fig. 7a) and years in which spring turnover occurred (Fig. 7b). Anoxia began earlier in spring in the "failure of spring turnover" years (Fig. 7a). The interannual variability in these distributions (Fig. 7) is likely attributable to variations in the stratification regime driven by natural meteorological variability. The distribution for post-closure conditions for the 1987-1992 interval (Fig. 7c) is clearly narrower than the pre-closure distributions (Fig. 7a and b). The polynomial lines fit by least squares regression (Fig. 7d) support this position. Thus the lower layers generally remained oxygenated longer in spring and early summer and were replenished earlier in fall in the 1987-1992 interval (Fig. 7d), consistent with the shorter period of stratification observed in those years (Fig. 4). This is also shown by the early onset of anoxia in spring of 1993 and 1994. In these two years, spring turnover failed to occur and the pre-closure DO pattern recurred (Fig. 7d). Observations in April of both years and May of 1993 track very closely the pre-closure pattern, and fall outside of the envelope of 1987-1992 observations (Fig. 7c). The conditions in late May 1994 (Fig. 7c) approach those observed over the 1987-1992 period (Fig. 7c), because of high levels of vertical exchange (Fig. 5b).

Management Perspectives

Ionic waste discharged by a soda ash/chlor-alkali plant made certain inflows more dense than the lake.

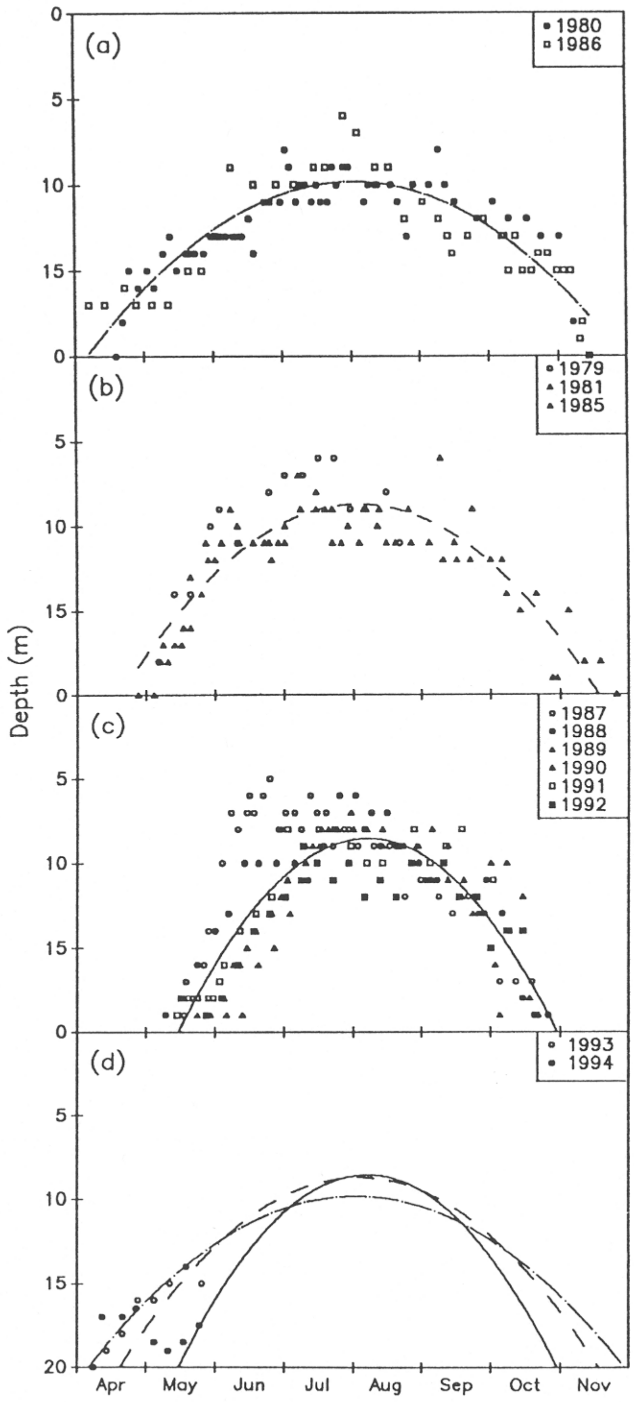

Figure 7.–Temporal distributions of the upper depth boundary of anoxia in Onondaga Lake: a) before closure of soda ash facility, case of "failure of spring turnover", b) before closure, case of occurrence of spring turnover, c) after closure, 1987-1992, and d) comparison of polynomial curve fits of distributions a) - c) and April and May observations of 1993 and 1994.

These dense inflows tended to plunge in the lake; at times the inflows plunged to the lake bottom. Thus, the soda ash/chlor-alkali facility had a profound impact on the stratification regime of Onondaga Lake during its operation. The lake annually had a substantial salinity component of stratification, the period of summer stratification was extended, spring turnover failed to occur in a number of years, and reformation of salinity-based density stratification was common in fall. These alterations of the stratification regime have exacerbated the lake's problem of degraded oxygen resources. Other potential ecological and water quality impacts from these alterations to the stratification regime should be evaluated.

The impacts on the stratification regime and oxygen resources were ameliorated by reductions in the salinity, and therefore density, of the inflows that resulted from closure of the soda ash/chlor-alkali facility. Lingering impacts continue as a result of continuing inputs of ionic waste from soda ash manufacture. Remediation of these impacts should focus on the major source of the ionic waste loading, the most recently active disposal area for soda ash wastes along one of the lake's principal tributaries (Effler et al. 1996).

References

American Public Health Association. 1980. Standard Methods for the examination of water and wastewater. 15th edition American Public Health Association, NY.

Bowie, G. L., W. B. Mills, D. B. Porcella, C. L. Campbell, J. R. Pagenkopf, G.L. Rupp, K.M. Johnson, P.W.H. Chan, S.A. Gherini and C.E. Chamberlin. 1985. Rates, constants, and kinetic formulations in surface water quality modeling (Second Edition). EPA/600/3-85/040, United States Environmental Protection Agency, Environmental Research Laboratory, Athens, GA 30613.

Chen, C. T. and F. J. Millero. 1978. The equation of state of seawater determined from sound speeds. J. Mar. Res. 36:657-691.

DiToro, D. M. and J. P. Connolly. 1980. Mathematical models of water quality in large lake, Part 2: Lake Erie, USEPA, Environmental Research Laboratory, EPA-600/3-80-065, Duluth MN.

Doerr, S. M., S. W. Effler, K. A. Whitehead, M. T. Auer, M. G. Perkins and T. M. Heidtke. 1994. Chloride model for polluted Onondaga Lake. Wat. Res. 28: 849-861.

Effler, S. W. 1987. The impact of a chlor-alkali plant on Onondaga Lake and adjoining systems. Wat., Air, Soil Pollut. 33: 85-115.

Effler, S. W. 1996. Salinity (in Chapter 5, Chemistry). *In* S. W. Effler (ed.). Limnological and engineering analysis of a polluted urban lake. Prelude to environmental management of Onondaga Lake, NY. Spring-Verlag, NY (in press).

Effler, S. W., C. M. Brooks, J. M. Addess, S. M. Doerr, M. L. Storey and B. A. Wagner. 1991. Pollutant loadings from Solvay waste beds to lower Ninemile Creek, New York. Wat., Air, Soil Pollut. 55: 427-444.

Effler, S. W. and R. Hennigan. 1996. Onondaga Lake, New York: legacy of pollution. Lake and Reserv. Manage. 12(1):1-13.

Effler, S. W. and E. M. Owens. 1986. The density of inflows to Onondaga Lake, USA, 1980 and 1981. Wat., Air, Soil Pollut. 28: 105-115.

Effler, S. W., E. M. Owens and K. A. Schimel. 1986a. Density stratification in ionically enriched Onondaga Lake, USA. Wat., Air, Soil Pollut. 27: 247-258.

Effler, S. W. and M. G. Perkins. 1987. Failure of spring turnover in Onondaga Lake. NY (USA). Wat., Air, Soil Pollut. 34: 285-291.

Effler, S. W., M. G. Perkins and C. M. Brooks. 1986b. The oxygen resources of the hypolimnion of ionically enriched Onondaga Lake, NY, U.S.A. Wat., Air, Soil Pollut. 29: 93-108.

Effler, S. W., M. G. Perkins, K. A. Whitehead and E. A. Romanowicz. 1996. Ionic inputs to Onondaga Lake: origins, character, and changes. Lake and Reserv. Manage. 12(1):15-23.

Effler, S. W., K. Schimel and F. J. Millero. 1986c. Salinity, chloride, and density relationships in ion-enriched Onondaga Lake, NY. Wat., Air, Soil Pollut. 27: 169-180.

Effler, S. W. and K. A. Whitehead. 1996. Tributaries and discharges (Chapter 3). *In* S.W. Effler (ed.). Limnological and engineering analysis of a polluted urban lake. Prelude to environmental management of Onondaga Lake, NY. Springer Verlag, NY (in press).

Field, S.D. 1980. Nutrient-saturated growth in hypereutrophic Onondaga Lake, Syracuse, N.Y. Ph.D. Thesis presented to Dept. of Civil Engineering, Syracuse University, Syracuse, NY.

Fischer, H. B. and R. D. Smith. 1983. Observations of transport to surface waters from a plunging inflow to Lake Mead. Limnol. Oceanogr. 28: 258-272.

Ford, D. E. and H. G. Stefan. 1980. Thermal predictions using integral energy model. J. Hydr. Div. ASCE 106: 39-55.

Harleman, D. R. F. 1982. Hydrothermal analysis of lakes and reservoirs. J. Hydraulics Div. ASCE 108: 302-325.

Hutchinson, G. C. 1957. A treatise on limnology. Vol. I. Geography, physics and chemistry. John Wiley and Sons, NY.

Jassby, A. and T. M. Powell. 1975. Vertical patterns of eddy diffusion during stratification in Castle Lake, California. Limnol. Oceanogr. 20: 530-543.

Lam, D. C. L. and W. M. Schertzer. 1987. Lake Erie thermocline model results: Comparison with 1967-1982 data and relation to anoxia occurrences. J. Great Lakes Res. 13: 757-769.

Martin, S. C., S. W. Effler, J. V. DePinto, F. B. Trama, P. W. Rodgers, J. Dobi and M. C. Wodka. 1985. Dissolved oxygen model for a dynamic reservoir. J. Envir. Eng. Div. ASCE 111: 647-664.

Millero, F. J., A. Gonzalez and G. K. Ward. 1976. The density of seawater solutions at one atmosphere as a function of temperature and salinity. J. Mar. Res. 34: 61-93.

Onondaga County 1971-1980. Onondaga Lake Monitoring Report Annual Report. Onondaga County, Syracuse, NY.

Orlob, G. T. 1983. One-dimensional models for simulation of water quality in lakes and reservoirs (Chapter 7). Mathematical modeling of water quality: streams, lakes, and reservoirs. *In* G.T. Orlob (ed.). Wiley and Sons, NY.

Owens, E. M. and S. W. Effler. 1996a. Modeling the impacts of a proposed hypolimnetic wastewater discharge on stratification and mixing in Onondaga Lake. Lake and Reserv. Manage. 12(1):195-206.

Owens, E. M. and S. W. Effler. 1996b. Hydrodynamics and transport (Chapter 4). *In* S.W. Effler (ed.). Limnological and engineering analysis of a polluted urban lake. Prelude to environmental management of Onondaga Lake, NY. Springer Verlag, NY (in press).

Owens, E. M. and S. W. Effler. 1989. Changes in stratification in Onondaga Lake, New York. Wat. Resour. Bull. 25:587-597.

Powell, T. A. and A. Jassby. 1974. The estimation of vertical eddy diffusivities below the thermocline in lakes. Wat. Resour. Res. 10: 191-198.

Stauffer, R. E. and G. F. Lee. 1973. The role of thermocline migration in regulating algal bloom. *In:* Modeling the eutrophication process. *In* E. J. Middlebrooks, D. H. Falkenborg and T. E. Maloney (eds.). Ann Arbor Science Co., Ann Arbor, MI. P. 73-82.

Stefan, H., T. Skoglund and R. O Megard. 1976. Wind control of algae growth in eutrophic lakes. J. Envir. Eng. Div. ASCE 102: 1210-1213.

Stewart, K. M. 1971. Other Physical Considerations. *In:* Onondaga Lake Study, Water Quality Office. Environmental Protection Agency, Project NO. 11060 FAE 4/71, 0-439-910. U.S. Government Printing Office, Washington, DC.

Wetzel, R. G. 1983. Limnology. Saunders Publishing, Philadelphia, PA.

Wodka, M. C., S. W. Effler, C. T. Driscoll, S. D. Field and S. P. Devan. 1983. Diffusivity-based flux of phosphorus in Onondaga Lake. J. Environ. Engr. Div. ASCE 109: 1403-1415.